Setting priorities for conservation: protected area effectiveness, management, and quality of governance

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LUOVA
Doctoral Programme in Wildlife Biology Research

Department of Biosciences
Faculty of Biological and Environmental Sciences

University of Helsinki

ACADEMIC DISSERTATION

To be presented for public examination with the permission of the Faculty of Biological and Environmental Sciences of the University of Helsinki in Biocenter 2 (Viikinkaari 5) auditorium 1041, on May 11th 2016 at 12 o'clock noon.

HELSINKI 2016

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ISBN 978-951-51-2067-0 (paperback)

ISBN 978-951-51-2068-7 (PDF)

http://ethesis.helsinki.fi

Unigrafia

Helsinki 2016

"There's no shortcut path to the truth."

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II **Eklund, J.**, Blanchet, F. G., Nyman, J., Rocha, R., Virtanen, T., and Cabeza, M. Contrasting spatial and temporal trends of protected area effectiveness in mitigating deforestation in Madagascar. *Manuscript*.

III **Eklund, J.**, Coad, L. and Cabeza, M. Protected area management assessments do not reflect effectiveness in deforestation reduction. The case of Madagascar. *Manuscript*.

IV Arponen, A., Cabeza, M., **Eklund, J.,** Kujala, H., Lehtomäki, J. 2010.Costs of Integrating Economics and Conservation Planning. *Conservation Biology*. 24: 1198-1204.

V **Eklund, J.**, Arponen, A., Visconti, P., and Cabeza, M. 2011. Governance Factors in the Identification of Global Conservation Priorities for Mammals. Philosophical Transactions of the Royal Society. Biological Sciences. 366: 2661-2669.

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Original idea	**JE**, MC	MC, **JE**	MC, LC, **JE**	AA	**JE**
Study design	**JE**, MC	MC, **JE**, GB	**JE**	AA, MC, **JE**, HK, JL	**JE,** AA, MC
Data		**JE**, JN, RR, TV	**JE,** LC		PV, **JE**
Methods and implementation / Analyses	**JE**, MC	**JE**, GB	**JE**	AA, MC, **JE**, HK, JL	**JE,** AA
Manuscript preparation	**JE**, MC	**JE**, MC, GB, RR, TV, JN	**JE**, LC, MC	AA, MC, **JE**, HK, JL	**JE,** AA, MC, PV

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ABSTRACT

Protected areas are a key tool for conserving biodiversity and an increase in their coverage has long been the aim of international conventions and initiatives. With progress to achieve target 11 of the Convention on Biological Diversity of protecting 17 % of terrestrial areas, the focus has now shifted towards assessing the protected area effectiveness in maintaining species or avoiding land conversions. In my doctoral thesis I develop a novel way of assessing PA effectiveness, based on the counterfactual thinking, and use this to link it to different management and ecological factors. I link different aspects of PA effectiveness conceptually to the quality of governance and show how spatial prioritizations can change with the inclusion of these socio-political factors.

Using Madagascar as a case study, and in line with other studies elsewhere, I find that the protected area network is effective to some extent in mitigating the pressure of deforestation. I show the importance of considering the temporal dimension of protected area effectiveness measures and how protected area effectiveness changes over time due to increasing or decreasing pressures. These results link directly to considerations of vulnerability and irreplaceability in Systematic conservation planning and I show that accounting for governance factors in a global spatial prioritization analysis change the identification of areas.

My thesis shows the relative nature of protected area effectiveness measures and how important it is to get the assessments right, especially because of the massive focus on protected area effectiveness as a panacea to stopping biodiversity declines. Improving protected area effectiveness needs to be linked to governance factors affecting not only the management but also the drivers of threat, something that previous studies have overlooked. With my thesis I make an attempt to bridge the themes of protected area effectiveness, considerations of quality of governance, and how it all links to conservation prioritizations.

Our methodology has been developed with the aim to be computationally efficient and conceptually more robust than existing matching methods, with the potential to be scaled up for larger studies. However, how the two methods perform needs to be tested in the future. My dissertation has clear practical implications for the conservation of Madagascar's biodiversity and the results are of potential interest for both NGOs and the Madagascar National Park administration. The conceptual contribution of this thesis should be incorporated into mainstream thinking and the discourse of setting global priorities for biodiversity conservation, such as by the International Union for Conservation of Nature and Natural Resources (IUCN), the World Parks Congress (WPC) and ultimately the Convention on Biological Diversity (CBD).

ABSTRAKT

Grundandet av naturskyddsområden är den ledande strategin för att skydda den biologiska mångfalden både lokalt och globalt. Stora framsteg har gjorts i frågan om att öka naturskyddsområdenas antal och areal och Biodiversitetskonventionens Aichi målsättning att skydda 17 % av jordens terrestra yta ligger inom räckhåll. Detta har gjort att fokus nu har skiftat till att man försöker evaluera naturskyddsområdenas effekt ifråga om att skydda de arter samt naturtyper som finns representerade. Analytiskt utgör detta ett intressant problem inom vetenskapen: det kräver en evaluering av vad som skulle ha hänt, om dessa områden inte grundats: det kontrafaktiska (från engelskans counterfactual). I min doktorsavhandling har vi utvecklat en ny metod för detta. Jag använder denna metod för att evaluera naturskyddsområdeseffektivitet och undersöker förhållandet mellan detta och olika aspekter på naturskyddsområdesledning, förvaltningsfaktorer samt ekologiska faktorer. Konceptuellt utvecklar jag ett ramverk som linkar olika aspekter av naturskyddsområdeseffektivitet till förvaltningens kvalitet (quality of governance) och visar hur spatiala prioriteringar beträffande var naturskyddsområden borde grundas kan ändras då dessa socio-politiska faktorer beaktas.

Genom att använda Madagaskar som fallstudie, och i enlighet med andra studier från andra länder, visar mina analyser att nätverket av naturskyddsområden har en effekt, det vill säga de minskar skogsskövlingen inom områdenas gränser. Dock påvisar jag vikten av att beakta den temporala aspekten av effektivitet, och hur den effekt områdena har varierar över tid och i förhållande till ett minskat eller ökat tryck på att skövla skog.

Min avhandling påvisar således den relativa karaktären av mått på naturskyddsområdeseffektivitet och hur betydande det är att genomföra dessa bedömningar rätt, samt hur resultaten skall tolkas. Detta framstår för tillfället som ytterst aktuellt på grund av det massiva fokus naturskyddsområdeseffektivitet har fått som ett universalmedel för att eliminera nedgångar i den biologiska mångfalden. Förbättrad naturskyddsområdeseffektivitet måste kopplas till förvaltningsfaktorer som påverkar inte bara den lokala ledningen utan även drivkrafterna bakom flera av hotbilderna mot biodiversiteten, något som tidigare studier har förbisett. Med min avhandling gör jag ett försök att syntetisera tematiken kring naturskyddsområdeseffektivitet och kvaliteten på samhällsstyrningen, och hur det hela länkar till hur man kan prioritera olika områden samt tillvägagångssätt inom bevarandebiologin.

Vår nya metod har utvecklats med målsättningen att vara beräkningsmässigt effektiv och konceptuellt mer robust än befintliga metoder, och att därmed ha potential att skalas upp för större studier. Hur de två metoderna utfaller kräver testning i framtiden. Min avhandling har praktiska tillämpningsmöjligheter för bevarandet av Madagaskars biologiska mångfald och resultaten är av potentiellt intresse för både medborgarorganisationer och administrationen för Madagaskars nationalparker. Det konceptuella bidrag denna avhandling medför bör införlivas i diskussionen kring hur man borde fastställa globala prioriteringar för bevarandet av biologisk mångfald, såsom genom de rekommendationer och mål som utges av den internationella naturvårdsunionen IUCN, världsparkkongressen World Parks Congress samt biodiversitetskonventionen CBD.

SUMMARY

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1 INTRODUCTION

1.1 THE LOSS OF BIODIVERSITY AND PROTECTED AREAS AS A SOLUTION.

Biodiversity (BD) is declining (Butchart et al. 2010; WWF 2014) and present extinction rates are estimated to be at least 1000 times higher than pre-human background rates (Pimm et al. 2015) and are predicted to increase markedly in the future unless drastic action is taken (Pereira et al. 2010). For the species with enough reliable data to assess, one fifth of vertebrate species are classified as “Threatened” and the red list indexes are deteriorating, with species moving closer to extinction (Butchart et al. 2004; Hoffmann et al. 2010, 2011; Juslén et al. 2013). What the effects of this loss will be remains to a large extent to be seen, but scientists warn of negative feedback mechanisms contributing to the deterioration of ecosystem services and even health effects on humans (Millennium Ecosystem Assessment 2005; Hanski et al. 2012).

Habitat loss and fragmentation due to human land use activities is presently the single most important driver of biodiversity loss (Millennium Ecosystem Assessment 2005). The pressures on biodiversity show increasing trends, with increasing impacts of human consumption, alien species, overharvesting, and climate change (Butchart et al. 2010). Global trends for habitat fragmentation are not available but are probably increasing as well (Butchart et al. 2010) which is likely to put species at higher risks of extinction than previously estimated based on species-area relationships (Rybicki & Hanski 2013). This thesis focuses on solutions related to mitigating the threats of habitat loss and fragmentation, but also touches to some extent on threats of direct overexploitation, such as hunting and trapping. The establishment of protected areas (PAs) has long been seen as a panacea to this (Hayes & Ostrom 2005).

Setting aside area to avoid land conversion is nothing new: starting from sacred sites and royal hunting forests, the establishment of PAs has a long history and undoubtedly reflects the prevailing ideas of the society, both moral and scientific, and not least reflecting the power relations of the time. Hence often the criticism from social sciences reporting exclusion, translocations, and illegalization of previously common practices of the local human populations (West et al. 2006). Nonetheless, PAs have increased globally, both in number and coverage, with a rapid increase since the 1970s (Juffe-Bignoli et al. 2014). Presently the IUCN reports 15.4 % of terrestrial and inland waters to be under protection. The situation is slightly lagging behind for marine areas, with only 8.4 % under protection, but in this thesis I will focus only on the terrestrial component.

Increasing the PA coverage has been a key target in international treaties such as the Convention on Biological Diversity (CBD), and most recently through the Aichi targets, setting the target of protecting 17 % of the terrestrial area globally (Juffe-Bignoli et al. 2014). In practice this has been interpreted to mean national targets and individual nations are now working towards reaching this goal (Woodley et al. 2012). This is despite the fact that scientists have repeatedly warned against establishing protected areas opportunistically (Pressey et al. 1993; Margules & Pressey 2000). Even though the coverage is increasing globally, many ecoregions remain under-represented (Butchart et al. 2015), and acting at the national level can be nonoptimal from the biodiversity point of view as it can lead to redundant measures (Pouzols et al. 2014). Previous research focusing on spatial aspects of PA networks has shown that many of them are too small to contain viable populations of key species (Oldfield et al. 2004) and that most PA networks over-represent economically marginal areas while failing to conserve many species and habitat types (Powell et al. 2000; Scott et al. 2001; Rodrigues et al. 2004; Joppa & Pfaff 2009).

1.2 SYSTEMATIC CONSERVATION PLANNING AND ADVANCED TOOLS FOR SPATIAL PRIORITIZATIONS

Systematic conservation planning developed as a response to the increased demand for more quantitative and systematic approaches to conservation. The opportunistic manner of establishing protected areas was criticized for not being based on key principles such as irreplaceability and vulnerability, and most notably complementarity (Margules & Pressey 2000; Cabeza & Moilanen 2001). Vane-Wright (1996) distinguished between two important questions: i) How to manage protected areas so that biodiversity is retained, and ii) where to locate the protected areas so that most biodiversity is protected? So far the latter question has received much more attention and developed into a research branch of *spatial conservation prioritization*, within the systematic conservation planning framework (Moilanen et al. 2009b). The fact that funding for conservation is limited made this new scientific branch take influences from economics. Presently the focus has shifted towards a more cost-effective approach, where economic costs are integrated into the conservation planning and clear and quantitative goals are set. Instead of valuing areas by their biological features only, they are also valued by the costs needed for conservation and this gives a priority setting where those areas able to conserve the most for the lowest cost will be valued highly. Different methodological approaches to solve these optimization problems have been developed. Complementarity is a key principle in reserve selection and closely related to the representation approach and the notion of efficiency (Pressey et al. 1993). Complementarity-based methods aim at giving the greatest combined species richness for a network of selected reserves.

The development towards integrating costs in the reserve selection process is reasonable as funding is always limited and we clearly will not be able to conserve everything. Moreover, it has been shown that the costs of conservation vary enormously—across 7 orders of magnitude per km^2—for terrestrial field based conservation (Balmford et al. 2003).This implies that integrating economic costs into conservation planning is very important and that return on investment analyses might actually be more dependent on differences in costs than in conservation benefits (Ferraro 2003; Bode et al. 2008). When applied at a global scale, this approach leads to prioritizing developing countries as the biological diversity is often high whereas land acquisition and management costs are low (Balmford et al. 2003). Attempts done at the regional level have however shown that high costs are likely to be associated with high levels of endemism or threat, and a too narrow focus on costs could mean that important areas for biodiversity remain unprotected (Moore et al. 2004). In addition to the distribution and alignment of costs and biodiversity benefits come the additional problems that emerge from this: developing countries suffer from severe underfunding for BD conservation (Balmford et al. 2003), and conservation in these areas will have to be carried out in a challenging socio-political setting, with problems of ineffective governance, high corruption, and fewer possibilities for civil society to be involved in conservation (McCreless et al. 2013). This could in turn jeopardize the effectiveness of protected areas.

1.3 HOW EFFECTIVE ARE ALREADY ESTABLISHED PAS?

Despite the fact that PAs remain the major policy instrument for biodiversity conservation, we know surprisingly little about their effectiveness in reducing threats and retaining biodiversity. Instead, there are numerous studies reporting continued habitat loss, populations declines, poaching, and encroachment within PA borders (Craigie et al. 2010; Porter-Bolland et al. 2012; Lindsey et al. 2013; Geldmann et al. 2013). While assessments are still limited, PA effectiveness has increasingly been the focus of research. This has happened separately at two fronts. First, international initiatives have developed assessments and standards for PA management effectiveness (Hockings et al. 2006), and second, researchers have tried to assess the ecological aspects of PA effectiveness (Gaston et al. 2006; Andam et al. 2008). Only a few attempts have been made to make connections between the two, and with often confusing terminology. In the following two sections I will briefly present the existing concepts, methods, and terminology. The individual chapters (I, II, III) will go more into depth with some of the problems linked to these previous assessments.

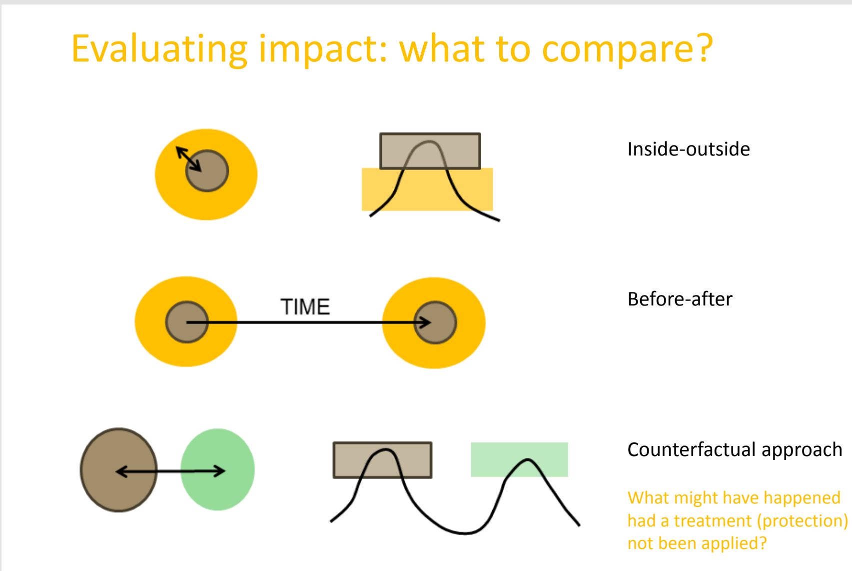
a) Protected area ecological effectiveness and conservation outcome

With the concept *ecological effectiveness* I choose to refer to changes in state of, or impact of PAs from a biodiversity point of view (e.g. changes in extent of forest cover, animal population trends). There is a substantial body of work investigating patterns of deforestation that has shown that forest loss occurs in many PAs, see Porter-Bolland et al. (2012) for a review of cases. Most PAs seem, however, to reduce deforestation rates compared to outside PA boarders (Sánchez-Azofeifa et al. 2003; Naughton-Treves et al. 2005; Nepstad et al. 2006; Geldmann et al. 2013), and this has often led to the conclusion that PAs are effective. However, PAs are often established in remote and less attractive regions (Joppa & Pfaff 2009), and recent studies that account for confounding factors linked to deforestation pressure, whether geographical, topographical, sociopolitical, or economic, have shown that protected areas may be less effective than previously thought (Andam et al. 2008; Gaveau et al. 2009a). Accounting for these other factors is vital because, given that many PAs are located in marginal areas, they are also under relatively less pressure. Consequently, this counterfactual approach, called “matching methods” (Andam et al. 2008; Stuart 2010), is increasingly used to measure PA outcomes, using the term *PA effectiveness* for their ability to reduce threat, ie. avoid land conversion. Studies in tropical parts of the world have consistently shown that PAs are effective (Andam et al. 2008; Gaveau et al. 2009a; Nolte et al. 2013; Carranza et al. 2014a), albeit accounting for the confounding variables reduce the perceived effect compared to simple inside-outside comparisons (see Box 1). However, these counterfactual studies are still relatively rare (Geldmann et al. 2013), computationally demanding, and almost completely lacking for assessments of species population trends, where it is difficult and expensive to survey outside of PAs (Craigie et al. 2010).

Finally, the threat reduction capacity of a PA requires good and efficient management but species persistence can also be the outcome of passive protection, and this can still be valid for the future of biodiversity. A PA can appear to be ecologically effective by virtue of good management or by virtue of low pressures. Differentiating between management and ecological aspects of effectiveness can be important in trying to see the underlying mechanisms affecting the outcome of PAs, and hence also how to improve the situation. This relative nature of effectiveness is important: a protected site with some deforestation in a context of high deforestation pressure may be more effectively managed than one with zero deforestation under no pressure (Nolte et al. 2013). The main questions still not addressed is how the PA effectiveness will behave under changing pressures. So far the scientific discussion seems to move strongly towards promoting these counterfactual assessments, regarding PAs with the highest threat reduction capacity as the most effective, without considering what will happen once the land conversion continues outside PAs and the pressures increase.

BOX 1. ASSESSING PA EFFECTIVENESS: FROM DIFFERENCES BETWEEN INSIDE-OUTSIDE TO THE COUNTERFACTUAL

Early studies aiming at assessing the impact PAs had on protecting biodiversity usually compared rates of land conversion, such as deforestation rates, inside PAs to outside areas, usually in buffer areas around the PA (Naughton-Treves et al. 2005). Another option, but rarely available for applied fields such as conservation biology, is temporal studies of before-after type, monitoring the changes taking place in an area after it has been set aside for nature protection. Both of these approaches have been criticized for having a skewed baseline for comparison and therefore risking overestimating the true effect of the establishment of the PA (Andam et al. 2008; Joppa & Pfaff 2010b). This is due to confounding factors, i.e. factors that are correlated with both the likelihood of protection and land conversion. PAs have been reported to be established in remote areas, often at high altitudes, where the pressures for land conversion are limited nonetheless, irrespective if the area is under formal protection or not (Joppa & Pfaff 2009). Accounting for these confounding factors is crucial when evaluating the impact a PA has, and the aim is to try to create a counterfactual scenario that estimates what might have happened had protection not been applied, and then compare the situation to this counterfactual setting. Assessing counterfactual scenarios have been used in other applied fields where it sometimes can be equally difficult to set up true experimental test designs, such as in epidemiology, sociology, and political sciences (Stuart 2010). These so called *matching methods* have been used to estimate the true effect of PAs, finding in general that PAs remain effective also after accounting for the confounding factors (usually altitude, slope, different distance metrics to infrastructure and commercial centers)(Andam et al. 2008; Gaveau et al. 2009a; Nolte et al. 2013; Carranza et al. 2014a), even if the effect is reduced compared to inside-outside comparisons.



b) Protected area management effectiveness

PA management effectiveness is receiving increasing attention in the conservation literature, mostly because of a concerted effort by international donors and NGOs to develop questionnaires for PA managers that assess threats, the local setting, and management effectiveness. Many of these assessments are based on concepts outlined in the management effectiveness framework developed by the IUCN World Commission for Protected Areas (Hockings et al. 2006) and includes elements of context, planning, input, process, output, and outcome. All these elements, including output and outcome, refer solely to the management process, and are not to be confused with the PA effectiveness or outcome considerations in the previous section. Examples include the RAPPAM methodology and the Management Effectiveness Tracking Tool (METT) (Coad et al. 2015).

These surveys have been undertaken in 90 countries (Coad et al. 2013) and analyses show that management in most PAs is "barely acceptable" (Leverington et al. 2008, 2010): 13 % are "paper parks" and lack any management activity while 62 % have basic management but with significant deficiencies (Leverington et al. 2010). Such studies have been criticized because they rely on managers, consultants, or government officials' responses and ratings based on own perceptions, which could produce biased results if respondents want to present positive outcomes (Ostrom & Nagendra 2006). Others argue that under the time and budget constraints, tools for quick evaluations based on expert knowledge are also needed (Hockings et al. 2009). While there are still too few accounts linking PA management effectiveness to ecological effectiveness, the nature of the management assessments may explain why recent studies from Brazil found no correlation between PA management effectiveness scores and reduction in fire occurrence (Nolte & Agrawal 2013) and habitat conversion (Carranza et al. 2014b). The PA management effectiveness assessments were originally developed to support adaptive management at the site or network level (Coad et al. 2015), and they are usually completed over a course of a few days by local managers and partners and sometimes representatives of local governments, local communities, or NGOs. More recently however, PA management effectiveness assessments have started to be used by funders for project evaluation purposes where project performance is measured as change in METT score, with the assumption that an increase in management effectiveness will have an effect on the biological performance /effectiveness of the PAs (Coad et al. 2015). This of course gives local managers a high incentive to report positive changes over time. However, if respondents are exaggerating about management effectiveness in their PAs, then this would make the findings of the global surveys (WWF 2004; Dudley et al. 2007; Leverington et al. 2008, 2010) of PA management effectiveness even more cause for concern.

This general low level of management effectiveness is particularly worrying because a recent study for tropical PAs found that the main predictor of "reserve health" is improved PA management (Laurance et al. 2012). Thus, there is a pressing need to understand what factors drive management effectiveness improvements. Such insights are available from national-level comparisons, which show that management effectiveness scores are much higher in countries with high or medium Human Development Index (HDI) scores (Leverington et al. 2008) and from studies showing that effectiveness at protecting biodiversity correlates with good monitoring and evaluation processes (WWF 2004; Dudley et al. 2007). Key activities identified are law enforcement and surveillance, strong links with regional authorities and local communities, and high institutional and governance capacity (WWF 2004; Dudley et al. 2007). Research has also found that PAs with lower management effectiveness scores tended to be those that are most threatened by over-harvesting (Dudley et al. 2007) and that PAs that are most effective at combating threats have the greatest support from political and civil society groups and higher levels of administrative effectiveness (Leverington et al. 2010).

In conclusion, the links between PA effectiveness and PA management effectiveness seem arbitrary and can stem from too high incentives to report positive outcomes, but it might also be that it is the result of factors that are beyond the influence of a local manager, such as national policies, government funding, governance, and development pressures. In order to give policy relevant recommendations it is crucial to separate between the part of management that can be influenced by managers and the aspects that will need to be addressed at different levels, this leads to considerations of the socio-politic and economic settings where PAs are established.

1.4 THE SOCIO-POLITICAL AND ECONOMIC SETTING WHERE PAs ARE ESTABLISHED

1.4.1 Institutional challenges

PAs are not established and managed in a vacuum, but within existing institutional arrangements and power relations, that is, within existing governance frames. Governance concerns the structuring of authority and setting of rules, and thus refers to how power is structured and how institutions are built, as well as how different institutions interact with each other. Various institutional arrangements, or governance systems, have been examined in relation to social-ecological systems (Ostrom 2007) and there is a wealth of literature on the importance of governance in determining various aspects of conservation outcomes (Brooks et al. 2006a; Chhatre & Agrawal 2008; Kenward et al. 2011). Coarsely categorized, there are four different types of PA governance: governance by government, shared governance, private governance, and governance by indigenous or local communities (Borrini-Feyerabend et al. 2013). Historically, PAs were often established and governed by the government, but more recently there has been a massive upswing in community approaches to PA governance (Balasinorwala 2014) as well as increasing attention to how the governance type affects the PA outcomes (Nolte et al. 2013). Focusing only on the governance aspects that are directly linked to PA type and management is not enough though. The list of potentially important factors is endless: education, livelihood options, land tenure, possibility to self-organize and affect decisions, to mention a few. As these factors cover several fields of science and society, I have decided to consider and test the usability of the concept of quality of governance as a simplified proxy for this multitude of dynamic feedback mechanisms at all societal levels.

My definition of *quality of governance* therefore focuses on the general policy environment within which institutions are framed or arranged. In doing so, it mostly refers to the control of corruption and transparency, political stability, the rule of law, and government effectiveness, but also aspects of equity and fairness (The World Bank group 2012; UNESCAP 2014). As such, some level of strong governance is required to produce a sufficiently stable policy environment in which to start building the institutions that determine the governance of a specific activity. The concept of *good governance* appeared in the 1990s as a way to measure the quality of governance, especially to inform decisions on where to focus development aid or business investments (Box 2). This link between quality of governance and effectiveness in achieving outcomes makes it particularly relevant for conservation.

Good governance has been linked to management effectiveness before (Lockwood 2010, Borrini-Feyerabend et al. 2013), but not to the conservation outcome concept discussed earlier. Lockwood (2010) suggests 7 principles of good governance in relation to PAs, namely legitimacy, transparency, accountability, inclusiveness, fairness, connectivity and resilience, and finally how the performance outcome of these could be measured. He also presents a framework for how *governance effectiveness* is linked to the PA management effectiveness framework by Hockings et al. (2006), recognizing that effective governance is a prerequisite of effective management. This framework by Lockwood is, however, not linked to the concepts of PA ecological effectiveness or to the counterfactual ways of measuring PA effectiveness, and this missing link remains an open question both empirically and conceptually, that I will address.

1.4.2 Funding and economic challenges

Conservation initiatives such as establishing PAs and maintaining them are costly. It is not simply a matter of acquisition costs and management costs, but also damage costs and opportunity costs (Naidoo et al. 2006), with the two last ones often being the burden of local communities affected by the establishment of PAs. Because the high priority areas for conserving biodiversity often are located in

BOX 2. GOVERNANCE AND AID

In general, richer countries have better governance and poorer countries suffer most from weak governance (Kaufmann & Kraay 2003). This is why the importance of quality of governance for achieving development outcomes has been an extended debate in the development sector. Studies suggest that aid will only lead to development in better governed countries (Burnside & Dollar 1997, 2004). This is based on the argument that better governed countries use aid money more effectively and therefore achieve expected development outcomes. Despite some controversy over the results and the role of governance (Easterly et al. 2003; Roodman 2007), aid effectiveness, and all the policy implications that it comes with, is now recognized as key in achieving development goals such as the Millennium Development Goals (Global partnership for effective development co-operation 2014). The increasing criticism that donor funds have not always been effective has led to debate on whether selectivity or conditionality should be used in allocating the aid. Conditionality involves setting policy conditions for aid (Dijkstra 2002), so that recipient countries have to carry out certain policies or reforms in order to receive the aid (ex ante). Selectivity, on the other hand, involves choosing which countries will receive aid and is therefore more a mechanism of rewarding good performance: a country has to perform first and only thereafter qualify for aid (ex post). Proponents argue that selectivity rewards those that deserve it, who then serve as role-models and motivators for states that are not selected. These ideas might not apply directly to conservation, but the links deserve to be acknowledged. Should conservation investments be directed to regions with favorable conservation conditions, or should biodiversity needs drive conservation priorities but effectiveness be enhanced in problematic regions in different ways?

developing countries (see section 1.2.), it is evident that the successful implementation of international conservation policies and targets such as the CBD will require major financial flows (James et al. 2001). A study looking at the official donor assistance for biodiversity during the past decades shows that the World Bank and the Global Environment Facility (GEF) stand for up to almost 60 % of total aid committed, followed by the United States as the biggest bilateral aid donor (Miller et al. 2013). The total aid sum since 1980 of US$ 18.55 billion still falls short of Rio commitments and Agenda 21 promises (Miller et al. 2013). Alarmingly, the 40 most severely underfunded countries contain 32 % of all threatened mammalian species, most of the underfunded countries being from the developing world (Waldron et al. 2013). Studies have found arbitrary results of how well allocated these international funds are in terms of protecting species or important BD areas (Halpern et al. 2006; Holmes et al. 2012; Miller et al. 2013). However, these assessments are mostly based on species richness accounts, and rarely based on the concept of complementarity, meaning that some of the conservation outcomes may be redundant, biased towards particular species and biomes. From a complementarity point of view, or aiming at covering all species globally, it is unknown how well allocated the limited funds are, and what potentially could be achieved with the funds invested. What remains likely though is that many PAs in developing countries strive with limited funds for implementing management actions in a challenging socio-political setting, a claim supported by the high level of so called paper-parks (Leverington et al. 2010). Interestingly, these aspects are largely ignored in PA effectiveness studies. Craigie et al. showed initial results for the impacts GEF funds have had for different PA outcomes, and seem to have found no clear relationship (Craigie et al. oral talk at ICCB 2015, see abstract book). Otherwise we are still largely lacking studies assessing the impact a PA makes relative to the budget it gets.

In conclusion, a key question not yet fully addressed is how the likelihood of success of a PA varies with funding and governance, and how this could be used to inform the setting of spatial priorities.

BOX 3. DEFORESTATION IN THE TROPICS AND IN MADAGASCAR

Tropical forests are crucial for the preservation of biodiversity, for providing local livelihoods, and for the role they play in mitigating climate change. Their role is recognized both in the convention on biological diversity (CBD), but also in the United Nations Framework Convention on Climate Change (UNFCCC), where policy tools for reducing emissions from deforestation and degradation (REDD+) are developed in order to mitigate climate change (Kanninen et al. 2007). Advances in satellite imagery and tools for analyzing the data allow for fine scale mapping of the state of these forests and changes happening through time. With the term tropical forest, I here refer to any of the forest biomes found in the tropics (between 23.5 degrees north and 23.5 degrees south of the equator), not only the humid forests of the Amazon basin, the Congo basin, Southeast Asia, and also Madagascar, but also dry forests and other types of savanna-like forest biomes (May-Tobin & Elias 2011). The tropical forests experienced increasing forest loss per year between 2000 and 2012, with Brazil as the exception (Hansen et al. 2013). However, the reduction in the deforestation rate of Brazil seem to have been offset by increased forest loss elsewhere, mainly in Indonesia, Malaysia, Paraguay, Bolivia, Zambia, and Angola (Hansen et al. 2013). Net deforestation in the African humid forests are also reported to have decreased between 2000 and 2010, compared to the period 1990-2000, but rates remain particularly high in Madagascar (Mayaux et al. 2013). Drivers for deforestation are usually analyzed though a framework of broad clusters of underlying driving forces underpinning the proximate causes such as infrastructure development, agricultural expansion, and wood extraction (Geist & Lambin 2002). The underlying causes are related to bigger societal changes and drivers, such as demographic, economic, and technological factors, but also governance factors and cultural aspects (Geist & Lambin 2002). Under the present era of global change, it is key to recognize that many of the underlying causes can be strongly linked to global markets and the demand for natural raw materials and products in societies far from the tropical forest itself (Boucher et al. 2011).

Addressing a multitude of the underlying drivers for land-use change seems crucial in achieving decreased forest loss. Even if economic factors (e.g. fluctuations in commodity markets) at first sight seem to explain changes in deforestation rates, the role of new policies and institutional arrangements governing those factors seem to have been significant (Mather 2007; Nepstad et al. 2014). In Brazil for example, expansion of soy bean plantations and cattle areas were the main drivers of deforestation. Consumer awareness and changes in supply chains such as restricting market access for deforesters in addition to enforcement of already existing laws, better cooperation between different ministries, and restrictions on access to credit at the county level seem to have been successful in reducing deforestation (Nepstad et al. 2014). This analysis is in line with previous research in forest transition theory that predicts that a country goes through a transition of first increased deforestation but then this slows down and policies supporting reforestation can even be implemented (Rudel et al. 2005). Some claim that a few Asian countries have reached the later forest transition phases with more forest cover remaining when compared with Europe and the US in the past (Mather 2007). In Africa, many countries are still at the early phase with diminishing forest frontiers, mainly driven by local scale agriculture, with potentially more industrialized logging yet to come (Rudel et al. 2009). For conservation biology, the inspection of forest loss and its main drivers in light of the forest transition theory and changing drivers could be useful; it could give a better understanding of the overall economic and policy environment and what role conservation actions can play in the wider landscape and at what institutional levels an input would most likely have a significant effect.

Madagascar serves well as a case study of a developing country struggling with tropical deforestation. It contains all the above mentioned types of tropical forest: humid, dry, and open bushlands (the spiny forest in the south). How much of Madagascar was originally covered by forest is an open debate, but instead of the often repeated 90 % -lacking rigorous scientific support- it seems likely that Madagascar has lost up to half (but perhaps less) of its primary forest cover in the last half of the twentieth century (McConnell & Kull 2014). Deforestation rates have been reported to decrease from 1990-2000 to 2000-2010 (Harper et al. 2007; Mayaux et al. 2013; ONE et al. 2013), with the differences most likely being explained by different methodologies, different resolutions, and different definitions of forest and forest loss (see McConnell & Kull (2014) for a review of the inconsistent results of deforestation rates in Madagascar). Despite most international focus going to the humid rain forests in the east of Madagascar, the rates of loss have been alarmingly high also for the spiny forest in the south-southwest (Harper et al. 2007; ONE et al. 2013)(chapter II). In Madagascar, as often elsewhere in the tropics (Boucher et al. 2011), the deforestation narrative has been simplified to poverty and population growth leading to widespread swidden agriculture and thus putting the blame mainly on the local farmers and subsistence agriculture (Scales 2014b). However, according to Scales (2014b) it is crucial to also consider other factors shaping land use, such as international or national policies and economic factors related to large-scale commercial agriculture. Already under French colonial rule, many policies promoting the government's desire to exploit the natural resources of the country were implemented, mainly in relation to timber and the cultivation of exportable cash crops (Scales 2014b). In this endeavor the rural subsistence farming was seen as a threat and a set of polices was developed to control it. In some regions, the implemented large plantations for cash crops not only cleared land, but also forced subsistence farmers further into the forest by taking up the most productive land areas; this process can be seen still today (Scales 2014b). In general, studies have identified a wide range of factors associated with deforestation, such as cash cropping by migrants, cash cropping by wealthy households, new roads, increases in international commodity prices, clearance for large scale plantations, and global political and economic factors (for a review see Scales (2014b)).

1.5 FROM GLOBAL TO NATIONAL: THE CASE OF MADAGASCAR

As the previous sections show, the global distributions of conservation costs, quality of governance, and priority areas for biodiversity show interesting patterns for developing countries (Fig. 1). This global level of analysis is of interest as a substantial amount of money for conservation originates in aid money but earmarked for biodiversity conservation, and thus with the potential to be allocated between developing countries (Miller et al. 2013). However, in order to look at some more local aspects, such as PA effectiveness and PA management effectiveness, I have included in this thesis case studies at a national scale, namely Madagascar (chapters II and III). Madagascar provides an interesting environment for the exploration of these questions. The country has among the highest level of endemism in the world, and has repeatedly been identified as a high priority for conservation (Brooks et al. 2006b), and is identified as one of the top 10 receivers of biodiversity aid (Miller et al. 2013). However, the country suffers from widespread poverty and more recently a very unstable political environment (Schwitzer et al. 2014), see also Figure 1. The change in presidential power after Madagascar's coup d'état in 2009 led to increased illegal loggings (Barrett et al. 2010; Innes 2010; Allnutt et al. 2013) and increased poaching and population declines of lemur populations in National Parks (Platt 2009).

Historically, Madagascar's PA network ranges back to the colonial times under French rule. The first protected areas were established in 1927, and the focus on establishing protected areas to combat deforestation continued in the independent state from 1960 onwards (Scales 2014a). The National Environmental Action Plan running from 1990-2009 served as the umbrella for the booming conservation activities of that time (Kull 2014), with the introduction of many integrated conservation and development projects, but PAs as still state governed. More recently, following the Durban Vision (with the goal of tripling the area under protection) and the change in political power since then, PAs under more sustainable use systems, such as community management, have been promoted albeit with little success in mitigating deforestation (Rasolofoson et al. 2015), the largest threat in the island. The effectiveness of the state managed PAs has not been assessed previously and as they still make up the majority of the protected lands, and a large share of the remaining forests, they deserve some attention.

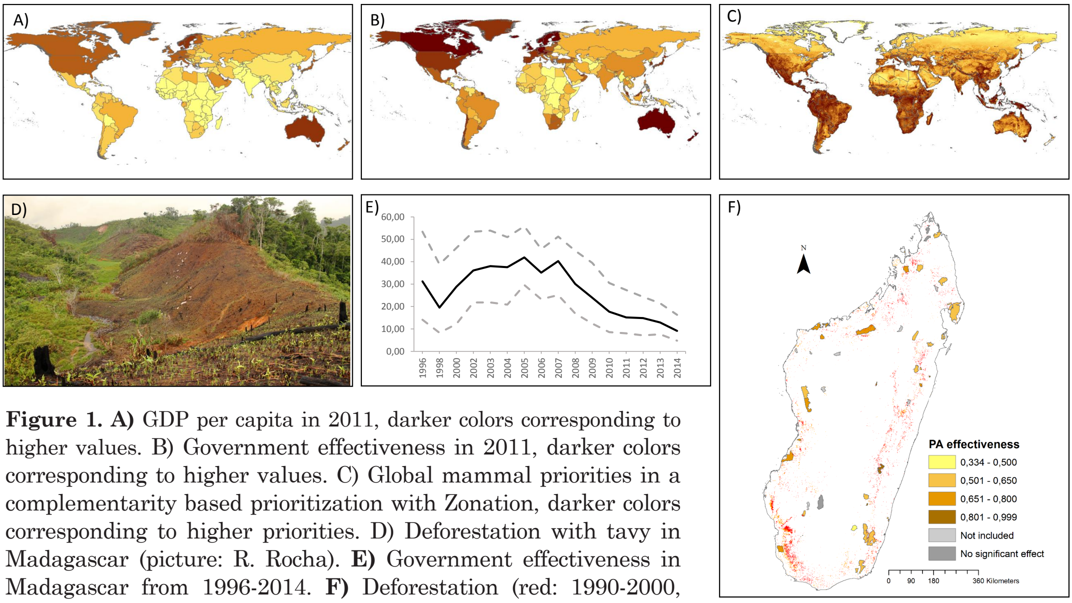


Figure 1. A) GDP per capita in 2011, darker colors corresponding to higher values. B) Government effectiveness in 2011, darker colors corresponding to higher values. C) Global mammal priorities in a complementarity based prioritization with Zonation, darker colors corresponding to higher priorities. D) Deforestation with tavy in Madagascar (picture: R. Rocha). **E)** Government effectiveness in Madagascar from 1996-2014. **F)** Deforestation (red: 1990-2000, orange: 2000-2010) and protected areas in Madagascar. PAs colored from yellow to dark brown, with darker colours corresponding to more effective areas (see Chapter **III**). For data see Table 1.

2 AIMS OF THIS THESIS

At the onset of this thesis, and as can be inferred from the previous introduction, it remained unclear how different aspects of PA effectiveness link to quality of governance considerations, while the focus remained merely on aspects of PA management factors. Methodological obstacles to address these issues appeared both due to undeveloped concepts and huge computational needs to perform analyses at relevant resolutions with representative samples sizes. Quality of governance in conservation planning was mostly ignored, whereas costs received increasing attention. Meanwhile the happenings in Madagascar, which had gone from being a model conservation nation and main receiver of BD aid with ambitious plans to increase the PA network, to collapse in a few months after the political coup, opened questions and considerations from global to national and local scales about what factors make conservation succeed. In light of these themes, I see room for further developing and problematizing the existing concepts of PA effectiveness, bridging these concepts to the field of conservation planning, and finally the need to integrate considerations of quality of governance as an overarching theme that should be addressed when integrating or bridging these fields. The main aims of this thesis are:

1. To discuss and evaluate different aspects of PA effectiveness

2. To both conceptualize and assess what management and governance factors are related to the above

3. To test how spatial priorities change if the above mentioned governance factors are accounted for

The thesis consists of five chapters/articles addressing one or a few of the above aims. Two of the chapters, chapter I and chapter IV, are more conceptual pieces of work, and the rest are empirical tests of some of the concepts coming out of these. Chapter I reviews the literature in relation to different aspects of PA effectiveness and links these through a novel conceptual framework to core concepts in conservation planning and aspects of governance. Chapters II and III both use Madagascar as a case study to empirically evaluate some of the ideas from chapter I. Chapter II focuses on assessing spatial and temporal trends in PA effectiveness, and for this purpose it uses a new methodology specifically developed to overcome some of the shortcomings in existing methods. Chapter III tests this methodology at the individual PA level and relates the effectiveness measures to aspects of management. Chapter IV is critiquing the development of the field of conservation planning at the time in blindly using cost or economic data to influence the spatial prioritization frameworks. The chapter was developed in parallel with chapter V, and influential in taking the step a bit further and testing some of this empirically. In chapter V I test how the inclusion of not only cost data, but also socio-political factors affect the identified priority areas. For an overview of the general aims and the specific research questions addressed in the included chapters see Table 1.

3 MATERIAL AND METHODS

3.1 DATA

Many different data sets were used in this thesis, see Table 2 for details of resolution and source and individual chapters (II, III, and V) for further description and exact references. The level of analysis ranged from global to national (Madagascar) and the chosen resolutions reflect this.

3.2 METHODS

The main strength of this thesis is in using and developing advanced methods to quantify concepts identified in chapters I and IV. For this the following three methodological approaches have been used.

3.2.1 Reviewing literature

Two chapters in this thesis are based on a review of the literature. The aim with chapter I is to synthesize emerging topics from three different fields: PA effectiveness, quality of governance, and conservation planning, into a conceptual framework broadening the present understanding of the matter by linking the three themes together. Chapter IV complements chapter V in discussing the potential caveats in including cost-data into conservation prioritizations without being aware of the implications it has at both the conceptual level and in practice. Both of the studies based on literature review have been developed adaptively with the more analytical chapters (Chapter I with II and III and chapter IV with V), with both processes influencing the development of the other.

3.2.2 Novel method for assessing the counterfactual

We developed a new methodology to assess protected area effectiveness based on the idea of assessing the counterfactual, i.e. what would have happened had the PA not been established? Existing so called *matching methods* have been used for the same purpose, creating an artificial control group, and then using point comparisons to matched pairs to estimate the true effect of protection (Andam et al. 2008; Stuart 2010).

The aim with this new method was to develop something better adapted to measure the effectiveness of protected areas than the commonly used *matching methods* because it compares each focal point to a set of similar points instead of a single "best" match. This can be especially important given the number of covariates, as points may be more similar for some covariates and not others. This is especially important also because matching is often done with replacement (Andam et al. 2008; Nolte et al. 2013), resulting in comparisons potentially done often to single points in the control group. Our method is developed to be computationally more efficient as it first partitions the environmental space and then searches within these partitions, instead of doing pair comparisons for each individual point (finding the closest pair from all possible pairs) and all other points in the sample. The idea with this partitioning, and the fact that our methodology allows for parallel computing, was to allow bigger sample sizes at finer resolution than what have been used in matching studies so far.

Our method uses the Mahalanobis distance on the set of covariate environmental characteristics to compare each focal pixel (all sample pixels from inside protected areas) to a group of pixels with similar characteristics. The pixels similar to the focal pixel are chosen following an iterative procedure. (1) First, we scale all the covariates and apply the Mahalanobis transformation on them. (2) The span of each transformed covariate is calculated thus giving the boundary of the full "environmental space". (3) Next, we divide the span of each transformed covariate by a predefined value (i.e. 20). This restricted span is then used to define a region around each focal pixel. (4) All pixels within this smaller cloud of pixels are defined as similar enough to use for comparison. Out of these, a user defined, preferably large, number of "most similar pixels" (i.e. 500 pixels) is chosen to make the comparison. (5) For all PA pixels that do not have 500 associated pixels, steps 3 and 4 are repeated but with slightly larger area to define the restricted environmental space (i.e. by diving the span of each transformed covariate by 19 instead of 20). This procedure is repeated until 500 pixels have been associated to each focal PA pixel, this cloud is referred to as the *similarity set*. For groups of pixels that are similar in the multidimensional environmental space, we can compute fractions that are protected and deforested and fractions that are non-protected and deforested, thus estimating the real effect of protection.

Quantifying effectiveness

Once the artificial control group, the so called *similarity set*, is developed as described above, the question becomes how to quantify the effectiveness. We compare difference in medians between the created PA group and the artificial control group and use this to quantify the effectiveness of protected areas in a way comparable to what is done with matching methods (e.g. Carranza et al. (2014a) on absolute effectiveness). We find this measure limited in its usefulness in our case and in addition to this compute effect sizes using PS_{dep} (Grissom & Kim 2012), a non-parametric effect size statistic that relates to the number of *similarity set*s indicating that protected areas are more effective than expected. In this thesis, I apply our methodology in two different ways. In chapter II, I apply it across all PAs for each forest type in Madagascar, and in chapter III, I apply it for individual PAs, accounting for the forest type. Matching methods have been applied similarly, but more often like the former, i.e. across a whole network of PAs (Andam et al. 2008; Carranza et al. 2014a).

3.2.3 Spatial prioritizations using Zonation

The tool used for identifying spatial priority areas for conservation was Zonation (chapter V). Instead of simply ranking areas based on their species richness or level of threat (Myers et al. 2000), Zonation is a complementarity based accelerated reverse stepwise heuristic (Moilanen et al. 2009a). The Zonation meta-algorithm starts from the full landscape and iteratively removes those cells whose loss causes the smallest marginal loss in the overall conservation value of the remaining landscape (Moilanen et al. 2009a). The marginal loss, i.e. the loss in conservation value when a cell is removed, can be defined in a few different ways, depending on the purpose of the prioritization. In chapter V the core-area Zonation was used for determining the conservation values, as I was specifically interested in retaining the representation of all species and not allowing for trade-off between species. The local biodiversity value of a cell is based on the species that has the highest proportion of its distribution remaining in the specific cell. In other words, the algorithm first removes cells with species that have wide distributions and aims at retaining equal amounts of habitat for all species. When a cost layer, as in chapter V, is used, cell

Table 1. The key research questions of the five chapters and how they contribute to the overall aims of this thesis

Chapter	Key research question	Geographical focus	Aim
I	What are the key aspects of PA effectiveness within a setting of drivers, pressures, state, impact and response?	Global	1,2, 3
II	Is the PA network in Madagascar effective in reducing the pressure of deforestation? How does the PA effectiveness change over time due to increasing/decreasing pressures?	Madagascar	1 and 2
III	How much does the individual PA effectiveness vary and is there a link to management factors?	Madagascar	1 and 2
IV	What are the problems with a too narrow focus on costs in conservation planning and what other factors should be considered?	Global	3
V	How does global priority areas change with the inclusion of both cost and quality of governance measures?	Global	3

removal is based on local biodiversity value divided by cell cost. I was specifically interested in investigating the effect of using cost versus quality of governance data on the identified global conservation priorities, and hence the proxies GDP per capita and corruption scores were re-scaled to vary between 0 and 100. Eight different conservation scenarios were produced, giving different weights to GDP per capita and corruption in the produced cost layer.

Zonation produces two main outcomes: the hierarchical priority maps (i.e. the rank priority maps) and performance curves, which quantify the proportion of the original occurrences remaining for each feature when successively smaller fractions of area of analyses remain in the process (Lehtomäki 2014). Both were analyzed in chapter V.

3.2.4 Computational limitations and resources used

Working with fine scale geographical information system data is computationally demanding. This applied to all my analytical chapters (II, III, and V), where the analyses were pushing the limits of what seemed feasible with a standard PC of the time. For Chapter V, Zonation v. 3 had just been developed to deal with that amount of data input. For chapters II and III, the methodology was developed and tested on an Intel (R) Xeon (R) CPU ES-26650 server with RAM 64.0 GB on 20 cores, but in order to more massively utilize the possibility of parallelization of the process, the final analyses were tested and run on the CSC Taito supercluster, allowing for parallel computation using an even larger number of cores (computational resources available for research by *CSC – IT Center for Science, Finland)*.

4 MAIN RESULTS AND DISCUSSION

In this thesis, I find that the PAs in Madagascar are effective in mitigating deforestation. My study shows both temporal and spatial variation, with the spiny forest appearing as the forest biome in most urgent need of attention due to high pressures to deforest

Table 2. List of datasets used in this thesis and in which individual chapter they were used.

Data	Description	Resolution	Source	Chapter
Forest cover	Forest layers that have been made by reclassifying the original land-use data based on ONE et al. classification of Landsat images.	30m x 30m	Conservation International Madagascar	II, III
Deforestation	Deforested pixels that are derived from the forest layers. The starting year's values are subtracted from the end year's values. Three different time periods, 1990-2000, 2000-2010 and 2005-2010.	30m x 30m	Derived from Forest cover	II, III
Distance to forest edge	Euclidean distance (m) to forest edge	30m x 30m	Derived from Forest cover	III
Distance to roads	Euclidean distance (m) to roads.	500m x500m	REBIOMA portal of Madagascar	II, III
Distance to major cities	Euclidean distance (m) to 4 major cities (of which two major ports).	500m x 500m		II, III
Distance to rivers	Euclidean distance (m) to rivers.	500m x 500m	Digital Chart of the World	II, III
Annual rainfall	Annual rainfall (mm) downscaled from 1 km to 500m resolution.	500m x 500m	WorldClim-Global Climate Data	II, III
DEM	Digital elevation data (m) from the Shuttle Radar Topography Mission.	90m x 90m	International Centre for Tropical Agriculture (CIAT), Consortium for Spatial Information (CGIAR-CSI)	II, III
Slope	Calculated from the DEM data in ArcGIS by using the slope function.	90m x 90m	Derived from above	II, III
Protected areas before 2000	Protected areas designated before or in the year 2000.	Vector	IUCN and UNEP-WCMC, The World Database on Protected Areas (WDPA)	II, III
Protected areas before 1990	Protected areas designated before or in the year 1990.	Vector	IUCN and UNEP-WCMC , The World Database on Protected Areas (WDPA)	II
Forest type	Forest type mask for three different classes: dry, humid, and spiny forest. The mask was digitized from reclassified vegetation data.	Vector	Reclassification based on the CEPF Madagascar Vegetation Mapping Project	II, III

Table 2. continued

Data	Description	Resolution	Source	Chapter
Management Effectiveness	Rapid assessment of PA management aspects, based on a scorecard questionnaire	For 36 PAs in Madagascar	Global Database on Protected Area Effectiveness	III
Mammal species distributions	Global distribution data based on habitat suitability models	0.1 degrees lat and long	Rondinini et al. (2011)	V
Control of corruption	Aggregated indicator on perception of corruption	Country	Worldwide Governance Indicators project	V
Government effectiveness	Aggregated indicator on perception of government effectiveness	County	Worldwide Governance Indicators project	I
Gross domestic product (GDP) per capita	Used as a proxy for conservation costs	Country	World Development Report 2009	V

also inside PAs (chapter II). I find no clear links between individual PA effectiveness and management effectiveness scores (chapter III). Chapter IV shows the caveats of a too narrow focus on including costs in conservation planning. Related to this, I find important trade-offs between cost and governance, and accounting for quality of governance in a global complementarity based spatial prioritization alters the regions selected (chapter V). In terms of protection levels, giving intermediate weights to both cost and governance seemed to perform almost as well as a prioritization based only on biodiversity (and thus serves as a good baseline to compare to) (chapter V). In chapter I, I present a conceptual framework linking the different aspects of PA ecological effectiveness, PA effectiveness, and PA management effectiveness to quality of governance at different levels. I build on this to present different routes available for integrating this into systematic conservation planning, specifying how concepts of vulnerability and irreplaceability are likely to interact with different strategies to allocate funding (selectivity vs. conditionality).

In the following sections I have chosen to highlight the most interesting aspects of my thesis, as I see it. The structure follows the main aims set in the beginning, but highlights merely some of the aspects, not all, and synthesizes between different chapters. For more in-depth discussion, the reader is referred to the individual chapters (I, II, III, IV and V).

4.1 PROTECTED AREA EFFECTIVENESS AND THE RELATIVE NATURE OF THE COUNTERFACTUAL MEASURES

Using Madagascar as a case study, and in line with other studies elsewhere (Andam et al. 2008; Nolte et al. 2013; Carranza et al. 2014a), I find that the PA network is effective to some extent in mitigating the pressure of deforestation (chapter II). The majority of the individual PAs are also effective (chapter III). I show the importance of considering the spatial and temporal dimension of PA effectiveness measures and how PA effectiveness changes over time due to increasing/decreasing pressures that also vary in space (chapter II) (Fig. 2). Figure 2 compares expected deforestation fractions for protected areas (yellow) and for overall forested pixels (green) while also comparing the effects of accounting for or ignoring confounding factors (blue line). It is clear that accounting for the confounding factors is really important, and

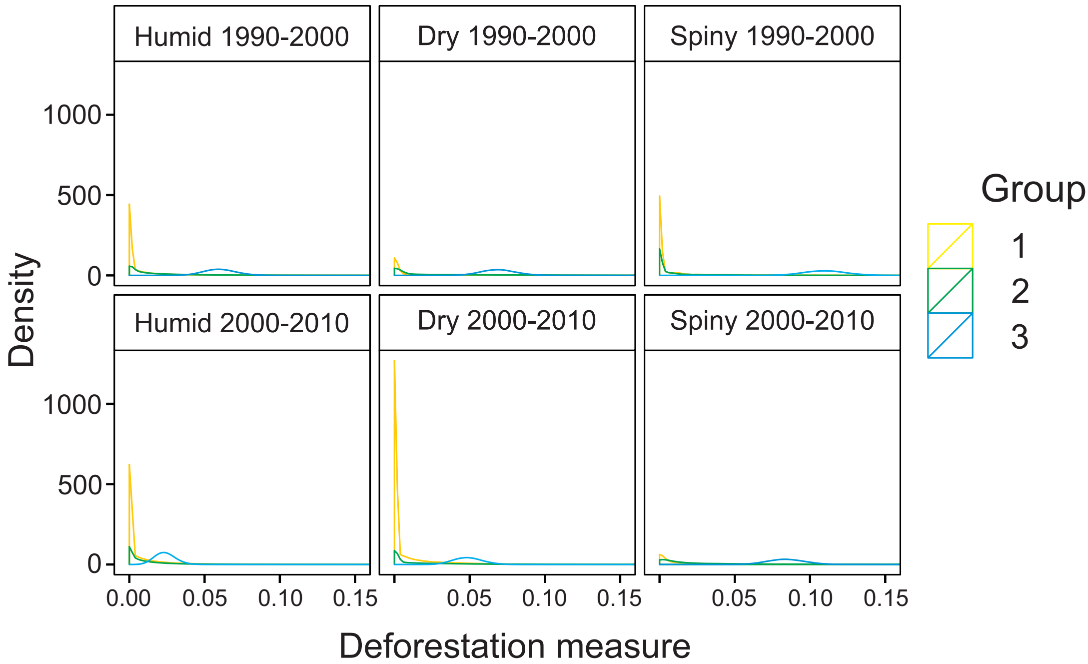


Figure 2. Density curves of deforestation fraction (fraction of deforested pixels out of a created 500 pixel similarity set). Adapted from chapter **II**.
1: Fractions of deforested protected pixels out of total protected pixels in the similarity sets.
2: Fractions of deforested pixels of the similarity sets.
3: Fractions of deforested pixels (disregarding similarities, disregarding protection, and other covariates).

they are indeed the main explanatory factors for the lower deforestation rates inside PAs compared to outside. Despite this, the protected area networks in all the three forest types do make a difference, even after accounting for the confounding factors, see Figure 3 and chapter II for more details.

Among studies quantifying PA effectiveness through the counterfactual, it is rare to consider the temporal aspects (but see Nolte et al. 2013; Haruna et al. 2014). As such, chapter II is an important advancement and pinpoints the danger of drawing conclusions too hastily: the way the artificial control group is dependent on the pressures outside PAs and hence PA effectiveness is dynamic and can increase or decrease depending on the context, both in terms of pressures and management. In chapter II, I explore these patterns in depth. For the humid forest, the PA effectiveness decreased in the second time period; this might be due to three reasons: a) deforestation has gone down overall (lower pressure, smaller role for PAs), b) deforestation is mainly happening in more easily accessible areas (lower pressures specifically in the areas that serve as points of comparison to PAs in the method), or c) the management of PAs is suffering and deforestation inside PAs is growing relatively more than in comparable areas. Looking at the values for the overall deforestation measure (Fig. 2), it becomes apparent that at least a) applies—the deforestation rate has decreased in absolute terms (blue line) and the deforestation in the matched baseline has also decreased (green line). Option c is improbable as the change in the deforestation inside PAs (yellow line) shows no sign of increase, except for the spiny forest in 2000-2010. For the spiny forest the deforestation in PAs has increased between the two time periods. However, at the same time, overall deforestation has gone down, but the deforestation in the PA-comparable areas has increased, indicating that the pressure on PAs has increased. The PAs in the spiny forest in the second time period have managed to mitigate them, even if there is an increase in deforestation also inside. The PAs in the dry forest follow the pattern of the spiny forest, with increased effectiveness in the second time period.

This result is easy to understand when thinking about the temporal trajectory of land use change. At first most forest loss takes place in easily accessible areas, and in a counterfactual approach PAs appear as relatively ineffective because they are hard to reach and so are the pixels chosen as their control group (i.e. yellow and green in the graph would both peak at low deforestation). Then as more and more forest is lost across the landscape, deforestation pressure extends to more inaccessible regions and PAs might come out as more effective if conversion is mitigated in protected sites, relative to comparable non-protected areas. Eventually, when land cover change has been so massive that also the pixels used for comparison to PAs have started to be deforested, there is the risk that PAs no longer have the capacity to mitigate this threat and the measure for PA effectiveness might start to go down again. In chapter II, I find indications of these different stages of land cover change, with the spiny forest standing out as an example of something already at the second stage, with the danger of moving towards the last scenario.

From asking what is effective to asking what does a proof of effect mean

Previous static assessments of PA effectiveness have also determined that PAs in general are making a difference, and through them land conversion is avoided (Andam et al. 2008; Gaveau et al. 2009a; Nolte et al. 2013; Carranza et al. 2014a). However, any attempt to assess the counterfactual through creating a control group from comparable land areas outside protection is very context dependent. Hence the only conclusion possible to draw has been that PAs make a difference and hence is not a wasted effort. In terms of prioritizing actions for conservation in practice, this is of limited use. What the temporal perspective or spatial comparison in chapter II may tell is whether changes in effectiveness seem to be due to changes in pressures or potentially more internal factors such as e.g. management. This is important information as it can give an indication for practical conservation recommendations about whether it will require changes in actions inside and/or outside PAs. Meaning that with a temporal and spatial counterfactual approach it is easier to detect the role that PAs are playing with changing pressures and whether improvements in management within, or larger actions (such as national scale policies to regulate pressures) are needed. This is very important in light of CBD's interest to increase PA coverage and improve PA effectiveness. As it is now, science can only tell that PAs are having an impact, but the counterfactual approach cannot make inferences about how the effectiveness will change with increased pressures (Haruna et al. 2014). The pattern for the spiny forest in Madagascar is illustrating this: in the earlier time period the effectiveness of the PAs in this region was the lowest, only to increase in the second time period, most likely as an effect of increasing pressures in remote areas that were used to create the control group. Note that overall the deforestation pressure had decreased also for this forest type, as for all the others. In a future where increased pressure for alternative land uses seems the most likely (Butchart et al. 2010), I think it is of utmost importance that scientist clarify that our presently used estimates of PA effectiveness is static and as our study (chapter II) shows they can change with the context. The temporal variation is especially important. A PA network that scores low in terms of effectiveness doesn't mean that it would not have the potential to increase in effectiveness, would the setting change. One aspect worth mentioning is that most previous matching studies have been made for so called middle income nations (Brazil, Cost

Rica, Indonesia)(Andam et al. 2008; Gaveau et al. 2009a; Nolte et al. 2013; Carranza et al. 2014a), not in countries suffering from as low levels of development as Madagascar. Another important aspect is that despite high deforestation rates in the Amazon and South East Asia (FAO 2010), much forest still remains intact, and in these counterfactual scenarios finding representative locations outside PAs to compare to is still possible. In the case of Brazil, it is also worth noting the increased efforts made by the government to control deforestation from around 2005 onwards (Nolte et al. 2013; Nepstad et al. 2014), clearly reducing the pressure on PAs and affecting their measured impact (Nolte et al. 2013). It would be crucial to combine the present methods to assess PA effectiveness with future scenario planning initiatives and in this way identify where the focus for management interventions and improvements in policies would be most crucially needed.

4.2 WHAT MATTERS FOR PA EFFECTIVENESS: LOCAL MANAGEMENT OR NATIONAL GOVERNANCE?

Building on chapters II and III, there seems to be no clear link between PA management and the PA outcome in terms of avoiding deforestation. This might seem like a surprising result, but a few previous studies have reported similar concerns (Nolte & Agrawal 2013; Carranza et al. 2014b). This lack of correlation can be due to two things: a) the proxy used for quantifying PA management effectiveness is not useful for these types of questions, or b) factors other than local management are more strongly related, such as national policies affecting the pressures and illegal actions, and thus overriding any potential effect local management could have. For example, the management effectiveness score used in chapter III has been designed for different purposes and is nowadays often linked to project evaluation and future funding decisions (Coad et al. 2015). This gives managers a pretty convincing incentive to report improvements in the management, regardless of the actual situation. In the case of Madagascar, the majority of PAs (29 out of 35) reported an improved management situation between the year 2005 and 2010 for which management effectiveness data is available. However, even if the differences between well managed and poorly managed PAs are not statistically significant, some interesting signals can still be appreciated in the comparison, such as that the poorest scoring PA in management effectiveness indicators also has what I refer to as induced deforestation, that is, higher rates of forest loss than expected given covariates. Higher than expected means that all other things being equal (elevation, accessibility, productivity) a forest inside a PA has more deforestation than a non-PA forest. This interesting phenomena has also been found for some PAs in the Brazilian rainforest (Nolte et al. 2013), and potential explanations could be local communities' hostile reaction to the PA, specifically targeting the protected land for forest cutting/clearing.

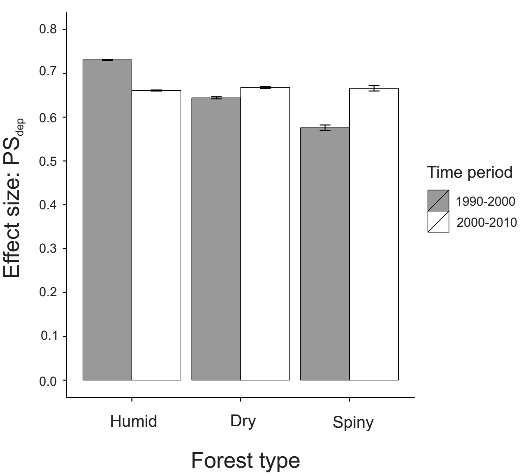


Figure 3. Changes in the effect size measure PS dep between forest types and time periods, whiskers show 95 % confidence intervals.

The finding that level of management cannot explain PA effectiveness fully gives some level of support for the framework developed in chapter I (see Fig. 4).

The framework in Figure 4 is an adaption of the DPSIR-framework. It links drivers, pressures, state, impact, and responses in a circular manner, favoring assessments of effectiveness that account for the interlinkages between aspects of society and the environment. In the framework, the different dimensions of PA effectiveness have been linked to the corresponding place in the causal chain. It identifies PA management effectiveness as an aspect of a RESPONSE, and the previously discussed counterfactual quantifications of PA effectiveness (avoided conversion) as the interaction between PRESSURES and STATE. However, in the framework, I leave room also for other aspects of PA effectiveness, such as more traditional inside-out comparisons, with the idea that a PA in an area under low pressure might still be effective under future scenarios of land-use change (see previous section 4.1). I claim that Drivers and Responses are very often interlinked (Fig. 4, panel B), such as in the case of quality of governance as the DRIVER and PA management as a RESPONSE. The framework might help in explaining the missing links between measures of PA effectiveness and PA management effectiveness. In conservation it is easy to imagine situations where local scale management interventions are rather powerless in halting extractive actions if the overall DRIVERS are too strong. In this thesis, I focus on quality of governance as one such driver, recognizing that other drivers include economic development, poverty, globalization etc. (see Box 3 in the Introduction). PA managers have only limited capacities/opportunities to influence any of these.

Interpreting the results from chapters II and III in light of this theory is useful and can explain the increasing pressures on the PAs in the spiny forest. Between 1990 and 2000, the southwest of Madagascar experienced a boom in maize cultivation that lead to a boom in the clearance of the spiny forest (Scales 2014b). According to Scales (2014b) there were many regional factors contributing to this boom, such as migration, lack of land-tenure, and lack of alternative livelihoods, but the main drivers were linked to international political and economic factors. Prior to 1990, maize had been cultivated mostly as a subsistence crop, but due to a change in EU policy, designed to stimulate economic development in the outermost regions of the EU (such as French Guiana, Guadeloupe, Martinique, and Réunion), Madagascar could start exporting maize to Réunion for its expanding pig farming (Scales 2014b). In addition to this, Madagascar had had to accept conditions of some structural adjustment programs by the International Monetary Fund, and remove trade barriers and open up for international commodity markets (Scales 2014b). This, in combination with the mentioned unclear land tenure and immigrant workers (Scales 2014b), has led to the vast areas deforested around Toliara that can be seen in Figure 1 F and the high deforestation pressure on PAs in this regions reported in Chapter II. Even if Madagascar's exporting to Réunion now has been outcompeted by bigger producers of maize, such as France and Argentina, the 1990s boom has had a lasting legacy and now the existing infrastructure allows for trading at the national market (Scales 2014b).

Examples outside of Madagascar also seem to highlight the role of economic drivers of deforestation and national policies to curb it. Previous studies in Indonesia have shown that local law enforcement is crucial but insufficient alone against increased pressures due to rising prices for agricultural commodities (Gaveau et al. 2009b). The recent success of Brazil in reducing its deforestation rate also gives some support for the massive importance nation-wide policies can have and highlights the importance of considering economic drivers in combination with different policies working at different levels (such as, in the case of Brazil, restrictions to market access for individual producers and/or producers in a county, better enforcement of existing laws, and improved communication and cooperation between different ministries) (Nepstad et al. 2014). I incorporated all of these aspects in my definition of quality of governance and how crucial this governance setting is in achieving conservation success. Another key aspect to consider is that PAs might vary in effectiveness in mitigating different threats, and separating between different drivers can be crucial, such as the difference between mechanized logging versus agricultural encroachments (Bruner et al. 2001; Gaveau et al. 2007; Rudel et al. 2009).

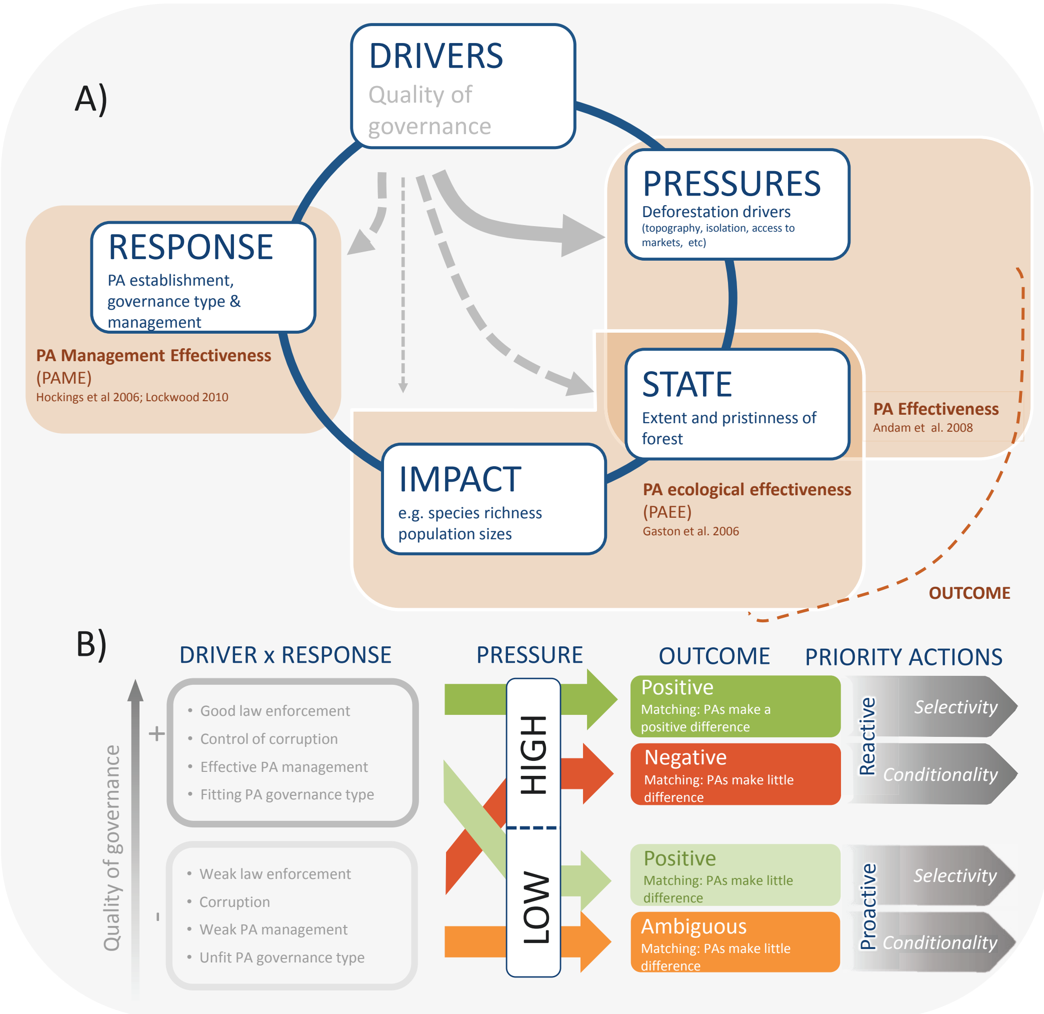


Figure 4. Framework identifying different aspects of PA effectiveness within a setting of drivers, pressures, state, impact, and response (panel A). In this thesis, quality of governance is considered as a driver and how it links to different measures of PA effectiveness is inspected. Panel B identifies the different combinations that can appear between different settings in driver × response compared to the pressures, and how this can result in different outcomes in terms of prioritizing action. See chapter I for details.

In conclusion, factors other than merely local PA management, such as the pressures for alternative land uses (chapters II and III) and potentially factors related to the original design of the PA network (such as size and fragmentation), are likely to determine whether a PA is successful or not. This suggests that the quality of governance, as a driver affecting many of these aspects, is a key component that has received too little attention so far in the conservation literature.

4.3 ACCOUNTING FOR GOVERNANCE FACTORS IN GLOBAL SPATIAL PRIORITIZATIONS

Assuming that the quality of governance affects both the PA effectiveness and the drivers of threat as suggested in 4.2 and chapter I, I wanted to explore the effect of accounting for them in a global spatial conservation prioritization. In chapter V we show how global priority areas change with the inclusion of both cost and quality

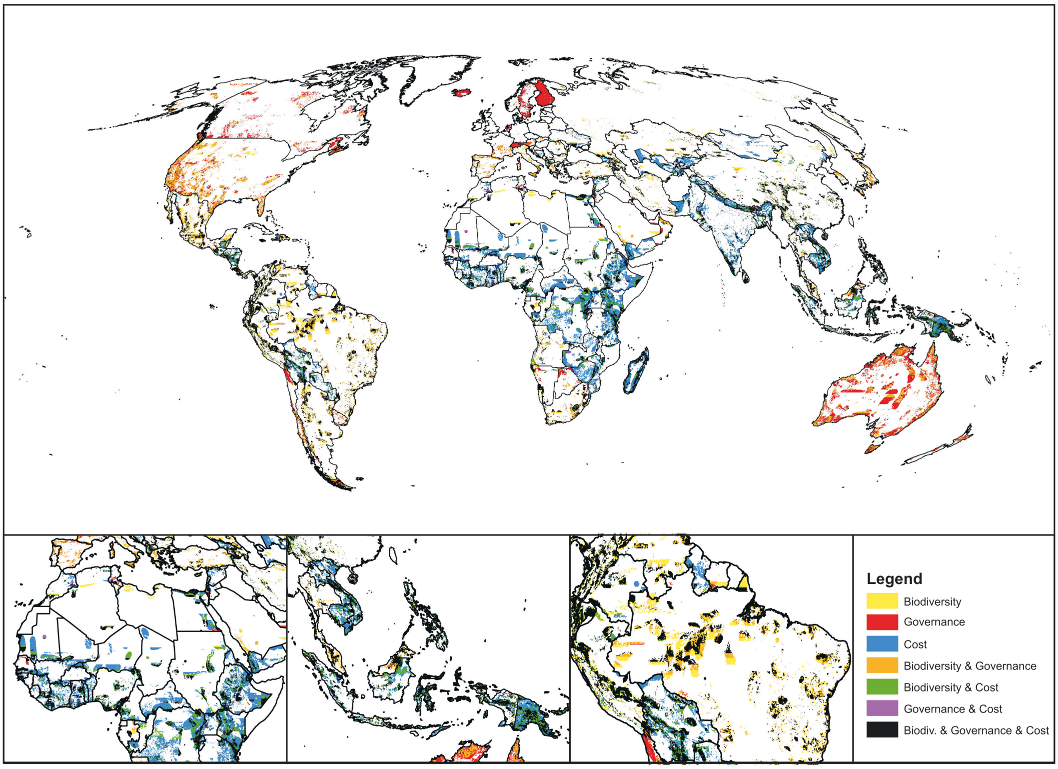


Figure 5. Spatial distribution of conservation priority areas selected in a complementarity based analysis with different scenarios. Top 10 % priorities from scenarios accounting for biodiversity only (yellow), costs only (blue), and governance only (red). Areas of overlap between different scenarios are shown in color blends. Adapted from chapter V.

of governance measures. Chapter V shows that, while core areas with high levels of endemism are always selected (a complementarity based approach to conservation planning), there are clear regional differences in selected sites when biodiversity, cost, or quality of governance is taken into account separately, see Figure 5.

Africa included many areas that stood out as priorities when only costs were considered, but not when only biodiversity was accounted for (areas in blue in Fig. 5). South America and Mexico had many areas that were important when accounting for biodiversity only, but due to higher costs in combination with lower levels of quality of governance, these areas were not selected in other scenarios (yellow). Parts of Europe, North America, and Australia stood out as important biodiversity areas with good governance (areas in orange in Fig. 5), but with high costs. With the highest weight for poor governance, some well-governed, but expensive and partly less biodiverse areas showed up in the prioritizations, such as part of the Nordic boreal region and Australia (areas in red in Fig. 5). Particularly parts of Africa and Southeast Asia showed large proportions of low cost areas important for biodiversity but which are suffering from weak governance (Fig. 5 in green). For these types of regions it would be crucial to follow a more *conditionality*-influenced approach to conservation (see Box 2 in intro), investing heavily in improving local management and capacity building, in order to achieve effective conservation.

This is especially alarming as solutions based on scenarios giving a high weight to economic costs also resulted in the largest overall corruption level within the top priorities, and these levels of corruption were much higher than the average corruption level across nations (Chapter V: Fig 2b). Those scenarios accounting for governance at the planning stage achieved substantial reductions in overall corruption levels. In terms of retaining species as percentage of protected land started to decrease, a prioritization based solely on biodiversity data performed best, with an economic cost scenario differing notably from all the others and starting to lose species coverage at larger fractions of area protected (Chapter V: Figure 3).

Chapter V is a simplistic approach to accounting for governance factors and lacks the validation of the strength the effect of quality of governance has on effective conservation. Because of this we explored different weighting scenarios in this chapter. Nonetheless, the idea can be conceptually linked to the framework in Figure 4 (chapter I). In accounting for quality of governance in prioritizing areas for conservation, the type of response would have to depend on whether a proactive or a reactive approach to conservation is taken (Brooks et al 2006b). Reactive approaches would address high pressure situations, with often a *selectivity* process that prioritizes high quality of governance and thus potentially effective outcomes in terms of PA effectiveness. If conservation targets require that a non-effective area is prioritized, this should be done under a *conditionality* process (see Box 2). Similar conditionality and selectivity approaches apply to proactive prioritizations, whereby conservation investments can be done in view of forecasts of threat. Low pressure, good governance areas could selectively be chosen, as potentially effective, whereas low pressure, poor governance areas should follow a conditional approach to become effective.

5 CONCLUSIONS AND FUTURE PROSPECTS

Even if biodiversity is declining (Butchart et al. 2010; WWF 2014), it seems like the main tools to address this decline is working: PA coverage is constantly increasing (Juffe-Bignoli et al. 2014), and PAs do make a difference, at least in reducing threats such as deforestation or general land conversion (Andam et al. 2008; Gaveau et al. 2009a; Joppa & Pfaff 2010a; Nolte et al. 2013; Carranza et al. 2014a; Chapters II & III). In terms of giving recommendation for policy making and practice though, it is important to consider where PAs are established (how representative is the network), and also how to interpret the results of PA effectiveness studies. This thesis stresses with strong emphasis the importance of considering the potential future changes to land use, where these will take place spatially, and how this will affect the temporal variation in assessments of PA effectiveness. Overall deforestation can decrease, and PA effectiveness measures can go down or they can go up, depending on where the pressures are situated spatially. It is now time to start finding answers to what makes a PA effective or not. An area can suffer from low levels of effectiveness because of poor management, because of high threats, or because of bad design in terms of size and connectivity, aspects that should be considered in future research.

One important question that I have not addressed in this thesis is the impact PAs have on local communities. In general there seems to be a prevailing idea that PAs affect local human populations in a negative way, taking away their possibilities for livelihoods and traditions (Brockington & Wilkie 2015). However, some interesting studies also using counterfactual scenario thinking have instead shown that PAs alleviate poverty (Andam et al. 2010; Ferraro et al. 2011). I have a true hope that future studies embracing multidisciplinarity and using best available methods from both social sciences and natural sciences could start scrutinizing this important question from different angles, perhaps looking at perceived poverty or victimization through participatory methods and using available datasets to assess counterfactual scenarios of what the situation might be had a PA not been established. Speculating of what would come out of these types of studies is useless, and certainly as always in conservation, the setting matters, so several case studies would need to be carried out before any conclusions can be drawn. However, as the state in the conservation biology community is now, sensing a certain division in ecologists and social scientists at for example the World Parks Congress 2014 arranged in Sydney, I fear that policy recommendations are made at highly influential levels without them being based on sound science. As such I recommend the IUCN, and especially the governance-branch there, to broaden the present focus on PA management and governance type, and to consider proper baselines for comparisons of the effect PAs have had on local people.

In light of the lacking correlations between PA management effectiveness and PA effectiveness (Nolte & Agrawal 2013; Carranza et al. 2014b, chapter III), I also raise the question of whether focusing on strengthening the management capacities in weak governance regions will be enough when the pressures seem to drive measures of PA effectiveness so strongly. Perhaps more *conditionality*-based approaches in challenging governance situations could be more directed to actions along the lines of the so called "new conservation"(Miller et al. 2011; Soulé 2013), i.e. investing in poverty reduction, education, capacity building, and providing alternative livelihoods, and the more traditional PA approaches to conservation could flourish in settings following a more *selective* approach.

Another question deserving more attention is the link between funding and PA effectiveness. With too limited budgets, a manager is unlikely to be able to make a big difference. Certainly one of the main reasons for this not having received much attention so far must be the lack of suitable and reliable data for these types of studies. However, considering the multitude of proxies, more or less suitable, used in conservation planning (Armsworth 2014) for costs it is nonetheless surprising. Many of the previously mentioned confounding factors (Box 1) that make protection correlated with extraction could also be correlated with potential sources of funding. This might be through direct project funding (Western researchers might be more likely to set up projects in areas of already existing projects and infrastructure). Another way is through tourism (Balmford et al. 2015), which is a major source of income for many PAs. In Madagascar for example, PA mangers expressed their frustration over the fact that the majority of tourists visit the parks with good infrastructure as a result of long term research projects (Eklund, *personal communication*). Thus hypothetically, we might end up with the "high and far" PAs as also underfunded, and more accessible areas as better funded.

However, despite the advances that this thesis makes to the topic of prioritizing actions for conservation, I have not managed to establish the link between quality of governance and success likelihood of conservation, such as for example measured through PA effectiveness. I have however, made an attempt to conceptualize the possible links at different levels, highlighting that quality of governance affects not only the management of a PA, but also the drivers and pressures for land conversion and other extractive uses of biodiversity. This will require a better consideration of the different scales at which conservation operates, ranging from international financial mechanisms, to national level policies and finally local implementation and management. Seen in light of this, it is not surprising that management effectiveness scores are not correlated with PA effectiveness, and perhaps the focus in conservation biology should start to move towards assessments of the links between governance factors and environmental and conservation policies implemented at the national level, and how these link to local management. Knowing that many of the needed datasets are already available, and that the new methodology I present in chapters I and II can be scaled up and used to assess these questions at the global level, I sincerely hope to be able to continue research along these lines.

6 ACKNOWLEDGEMENTS

I want to start by stating that this has been a marvellous journey. And like all good journeys, with its ups and downs and most importantly, with the main traveller coming out as changed at the end of the journey. Since I was a small child I have wanted to become a researcher in ecology. I sincerely want to thank everyone that has been part of making this dream come true. Mar - thanks for giving me a true challenge. I could not have found a better supervisor. I started off with a clear vision, thinking it's just a matter of getting the work done. But you took me on a journey to the unknown, to the place where I did not know what to do and how to do it. And oh, how I have enjoyed it! First desperate, lost, and unwilling to accept the fact that this time I will not get away so easily, but then slowly realizing what science is about, taking risks, trying different things, making mistakes, changing routes, trying to keep pushing forward. Mar, I admire you for your high research ethics and your drive to do good science that matters. Every single time we have met up, you have struck me with your sharpness and your quick associations, and this in combination with your very humane side makes you rather unique to work with.

I also want to thank Anni Arponen, who I have felt has been like an extra mentor for me. Thanks for all the sharing of ideas and discussions on most aspects related to doing science! I also want to thank you for all the extra work in relation to Chapter V that fell upon you when I was on maternity leave and unfortunately not at my sharpest.

I want to thank the members of my thesis support group: Eeva Furman, Markku Kanninen and Lotta Sundström. I have enjoyed our yearly meetings and the comments I have got from you. Lotta, thanks also for giving me the chance to gain some teaching experience during these years and for additional mentoring in building a career in academia.

I am grateful for the work done by the two pre-examiners of this thesis: Markku Kanninen and Lucas Joppa. I enjoyed reading the material you suggested and hope it has given me a better synthetic understanding of my topic.

I have been lucky to have such great co-authors for the chapters in this thesis. Thanks to Guillaume, for all statistics and coding, to Tarmo for helping me getting started with ArcGIS, to Ricardo for great discussions and collaboration, to Johannes for entering the project when it was at a critical stage and I was close to despair and somehow magically you managed GIS layers and Linux codes so quickly. Talking through some ideas with you have helped me clarify many things in my own head and you are just simply great to work with. Thanks also for designing the beautiful cover. Thanks to Piero for getting the chance to work with the mammal data. Lauren, thanks for inviting me to Oxford, and for collaborating with us on the PAME data. Lastly, Anni, Mar, Heini, and Joona, it was great to work with you all on the cost-essay, an idea that actually came out of our journal club discussions.

Carrying out this work in the Metapopulation Research Centre (MRC) has been marvelous. I want to thank Ilkka Hanski for the stimulating environment he has created, for the pursuit of truth, for the fact that I smile when I come to work and for the fact that I get to work with all these great people. Working in this environment has been beyond my expectations and there is all the time something super interesting going on. Ilkka, you make dreams come true. For so many of us. In my wildest dreams, I never thought there could be such an inspiring working place!

I want to thank the doctoral programme in wildlife biology research (LUOVA) and its coordinator Anni Tonteri. Anni, you have always been helpful and encouraging. There is not a single issue you have not found a solution to. People like you make the process of doing a Ph.D. so much easier.

The same applies to the great MRC personnel over the years: Sami, Viia, Marika, Aija, Johanna, Jenny and Kaisa. You have sorted out all kinds of bureaucratic stuff and I promise, I always tried to read your instructions before I came to ask for help. The frequency of how often that nonetheless happened goes to show

how vital you are for running this group! Thanks Sami for the layout of this thesis and your patience with my delays.

Within the MRC my closest reference group has been the people in Mar's Global Change and Conservation team: Anni, Heini, Erin, Marissa, Aili, Laura, Henna, Silvija, Ricardo, Antti, Álvaro, Annika, Piia, Helena, and Johannes. Thanks for the great company and all the peer-support. I also want to thank Marissa and Aili for great team work in Madagascar. In addition to these people I also want to extend my thanks to our biweekly journal club with pulla (Finnish buns)(with many of the people listed above and in addition to them Joona, Victoria, Aija, Andrea, Daniel, Peter, Ninni, and others). I have gained a lot of insight from discussing papers with you and criticizing them. I also want to thank Joona and Peter for helpful discussions on counterfactual scenarios, Enrico for talking conservation, Panu and Torsti for coding advice, Atte for help with Zonation and computational power, Anna-Liisa for getting the chance to participate in Science goes Kindergarten, and simply all MRC members for all the great science you do, for friendly smiles, and great times.

For maintaining my mental health through this process I am also indebted to my great office mates over the years, and I especially want to mention the support I have got from the other PhD-mums Anniina, Aija and Hanna. I remember early mornings (I got an early bird baby), tired, desperate, pushing the pram through snow to kindergarten, leaving a crying child and rushing to work, arriving with a bad conscience for not having enough time for either my child nor my work. Those mornings your kind words and 100 % understanding was vital! In addition to this, Victoria, thanks for always being there for me, for surprise chocolates when working late, and for taking the time to comment my English in this summary chapter (the faults in this section are my own). Tanjona, thanks for the ice cream and all the discussions and your mathematical approach to solving things (including life).

Mum and dad, thanks for always reading to us when we were small, for taking me to the library, for spending so much time with us in nature, and for waking up Saturday mornings to drive me for shoveling elephant dung. I remember with fondness our dinner conversations ranging all kinds of topics. Dearest sisters, you are my best friends and meeting with you usually end in super interesting discussions and all of us so excited we just shout out our opinions of how to improve the world. I am so happy I have you in my life!

Dearest Adrian, thanks for all the coffees, all useless pieces of cool information, all discussions, all smiles and all the happiness. For all our heated debates over conservation and land use change that slowly and reluctantly have made me realize that I want to bridge further with my research. Without you I would not have learned so much in this process, you are the most open-minded and non-judgmental person I know. Always keep questioning me, always keep arguing with me!

Emil, I especially remember the day you climbed your first tree. I stood there next to you, following how you set the goal in your head, how you carefully tested the branches for if they will hold, for how you coordinated your hands and feet, and your persistence at the task. You stopped, you reflected, you tried something different, you repositioned your hands and feet. And you made it. You were so happy, you shined like a sun. And I thought, very well done, now let's move on. But you climbed back down and then you did the same thing 20 more times. That's when it struck me what learning is all about. During all these years, following you growing and learning new skills has been the main inspiration for me. Thanks to you I have found the child in myself again, and the euphoria of learning new things.

I am grateful for the financial support I have got for this work. The LUOVA fellowship has been crucial in allowing me to focus fully on the thesis. I also thank the Academy of Finland (through the Finnish Centre of Excellence in Metapopulation Research), the Ella and Georg

Ehrnrooth foundation and Waldemar von Frenckells stiftelse for funding. I thank the Chancellor and LUOVA for travel grants for conferences. It feels incredible that someone has wanted to fund me to do exactly what I want to do.

I also feel privileged having done my thesis in a country where I have had access to information and knowledge, and I want to acknowledge the importance of all online subscriptions to journals that are easily taken for granted. When our computer analyzes turned out to be much more demanding than we had anticipated, I realized just how privileged I am. I feel grateful for the services provided by CSC – *IT Center for Science,* without your superclusters and computational power this thesis would not have been written. This means in the end I thank the tax payers of Finland. And since I am at it, I also want to thank the Finnish school system and the curricula. Free education is a privilege and I have been very lucky. Thanks to all my previous teachers, you have taught me to think, you have given me the tools to grow.

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